

Seasonal dynamics of waterbirds from a relict wetland in the central Monte Desert, Argentina

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Abstract

Wetlands currently have high rates of degradation, with more than 70% lost globally. In the central Monte Desert, Argentina, they are a scarce and limited resource for the biodiversity which depends on them. Waterbirds have been used as biological indicators of wetlands because they respond to fluctuations in food resources and to environmental changes in the short term. Here we analyse the seasonal variations in the structure of the waterbird assemblage from a relict wetland in this region. We carried out censuses of waterbirds in a 6-year period (between 2009 and 2019) during the southern summer and winter. We recorded 1875 individuals of 33 species of waterbirds during the summer and 677 individuals of 29 species during the winter. The grouping patterns of the waterbird assemblages differed between seasons ($R = 0.35$; $p < 0.01$). Taxonomic diversity profiles showed greater diversity

for all indexes ($^{\circ}\text{D}$) during the summer. The guild of invertivorous and omnivorous waders had a greater abundance of individuals during the summer ($p < 0.05$) and, together with the surface-feeding herbivores, contributed to the 87% of the dissimilarity of the assemblages between seasons. *Phoenicopus chilensis* was the only species registered as threatened with national and international extinction. Relict wetlands, such as Laguna del Viborón, still have attributes of community diversity and represent the last refuges for waterbirds of the central Monte Desert. The information gathered in this study will contribute to the guidelines for integrated management plans and monitoring programmes for the conservation of the wetland and its biodiversity.

Keywords

Argentina, biodiversity conservation, guilds, Laguna del Viborón, non-passerines, seasonality

Introduction

In arid and semi-arid areas, the scarcity and variability of the water resources condition the settlement and development of aquatic organisms (Castañeda et al. 2005). Therefore, the wetlands of these areas are highly variable ecosystems with a great diversity of organisms in an environmental matrix of less diversity (Sabo et al. 2005; Minckley et al. 2013). These environments are significant in the hydrological and ecological processes taking place within them, such as water storage, nutrient cycling and retention, climate regulation and refuge for biodiversity (Bezabih and Mossisa 2017; Serran et al. 2018). Wetlands are estimated to provide 40% of the ecosystem goods and services of all ecosystems worldwide (Costanza et al. 1997). Despite their ecological and socioeconomic importance, wetlands have been significantly declining in size and quality globally (Bolpagni and Piotti 2016; Young et al. 2016).

Agriculture is the world's largest user of freshwater and the main factor in degradation of surface and groundwater resources (Baigún et al. 2008). Changes in land use by agriculture degrade habitat quality and influence the hydrological cycle of wetlands physically and chemically (Asselen et al. 2013; Sica et al. 2016). Thus, there are changes in the species composition, abundance and distribution, resulting in changes in the community structure (Rayfield et al. 2009; Hagen et al. 2012). In the central Monte Desert of Argentina, the development of irrigated agriculture produced loss and degradation of wetlands (Prieto et al. 2008; Benzaquen et al. 2017). This is mainly due to the use of water for oases (i.e. areas cultivated by artificial irrigation) and the weather conditions which increase the water deficit in the Central Andes (Masiokas et al. 2019). Although the total loss rate of the wetlands in this region is unknown (Benzaquen et al. 2017), it was found that for the Ramsar site named Lagunas de Guanacache, Desaguadero, y del Bebedero, there was a loss of 80% of the coverage in the last 30 years at a rate of $-5.9 \text{ km}^2/\text{year}$ (Abrahan 2019).

Waterbirds include several species which have a strong influence on the structure and dynamics of wetlands (Andrade et al. 2018; Michel et al. 2020). This group plays a key role in controlling the populations of the lower trophic levels, influencing the energy flow and the nutrient cycling (de Arruda Almeida et al. 2019). Birds are the group of vertebrates in which the greatest spatial movements of species and individuals take place (Wiens 1989). The high mobility of birds allows them to

respond quickly to environmental fluctuations, such as those associated with annual cycles (Rubenstein and Hobson 2004; Jahn et al. 2020). Therefore, they maintain connectivity amongst wetlands by using watercourses as biological corridors for their movements and dispersal (Naiman et al. 1993). The structure of bird assemblages is directly associated with spatial and temporal variations in the availability, abundance and diversity of food resources (Marateo and Arturi 2013). Moreover, at a local scale, it depends on the habitat characteristics (e.g. wetland size and depth, vegetation heterogeneity and structure) and the availability of sites for resting or reproduction (Ma et al. 2010). In this way, each type of wetland can be described in terms of its characteristic avifauna and its composition will depend largely on the environmental attributes of the site. Despite their importance to the ecosystem, waterbird populations have been declining globally in relation to the degradation of wetlands (Kingsford et al. 2017; Amano et al. 2018).

Taking into account the value of wetlands for arid areas and the current high degradation rate (Minckley et al. 2013), there are scarce studies on the seasonal dynamics of waterbirds in the central Monte Desert (Contreras and Fernández 1980; Blendinger and Alvarez 2002). Bird species are used in conservation programmes under the assumption that the responses of individual species may represent the response of other taxa within the community, such as those belonging to lower trophic levels (e.g. invertebrates and fish; Vanni et al. 1990; Caro and O'Doherty 1999; Ikin et al. 2016). Bird species can respond independently to environmental variation and the presence or absence of one of them can show specific ecological characteristics of the habitat (Canterbury et al. 2000). The aim of this study is to analyse the structure of the waterbird assemblage of the wetland Laguna del Viborón, Argentina, during two seasons (summer and winter). Thus, we determined the assemblage of waterbirds, based on the species and guild diversity according to their diet and foraging behaviour. Nowadays, it is essential to quantify diversity in order to develop integrated management plans and make decisions in favour of the conservation of waterbirds and their habitats (Ma et al. 2010; Moreno et al. 2018).

Methods

Study area

The research was carried out in Laguna del Viborón, Mendoza, Argentina (32°53'18"S, 68°36'44"W). This site is part of the wetland system of the Leyes-Tulumaya streams (Rubio 2018). The Lake has a maximum surface area of 38 ha during the austral summer season and it shows variations throughout the year, mainly during the periods of water deficit (winter). It has an arid to semi-arid climate, with an average temperature of 25.4 °C in summer and 7.2 °C in winter and an average annual temperature of 16 °C (Rubio 2018). The average annual rainfall is 250 mm, concentrated in the summer. The hydrophilic vegetation is made up of cattails (*Typha domingensis*), Southern Bulrush (*Schoenoplectus californicus*, *Juncus* spp.), Common Reed (*Phragmites australis*) and Pampas Grass (*Cortaderia* spp.) (Benzaquen et al.

2017). The surrounding vegetation consists of shrub plants, xerophytes and halophytes, with plant associations characterised mainly by *Cyclolepis genistoides*, Seepweeds (*Suaeda divaricata*), *Allenrolfea vaginata* and Saltbush plants (*Atriplex* spp.) (Burkart et al. 1999).

The current complex of wetlands of the Leyes-Tulumaya system is one of the last relicts of the great lacustrine extensions of the central Mont Desert (Rubio 2018). This area was known as the Ciénaga del Bermejo in the eighteenth century and it was drained after the construction of channels and drainage in order to prepare the lands for agricultural production (Prieto et al. 2008). Nowadays, few wetlands make up the Leyes-Tulumaya system (Prieto and Rojas 2012). Water fluctuations of this system are subject to the hydrological cycle of the Central Andes and to the use of water in the agricultural areas of the northern oasis of Mendoza. Moreover, it receives water surplus from the rain run-off of the urban area, as well as from industrial and agricultural wastewater. The Leyes-Tulumaya system is an area with a high anthropogenic impact and does not have any environmental protection laws (Rubio 2018).

Bird survey

The samplings were carried out between 2009 and 2019 during two seasons: summer (February) and winter (July). The censuses were conducted by tracing a line-transect of 1.2 km in length per 100 m strip width. In this way, 12 censuses (6 in each season) were carried out, corresponding to six years (2009–2011 and 2017–2019). The transects were traversed at a constant minimum speed and were carried out during the morning (between 7 am and 10 am) in order to coincide with the highest peak of bird activity (Bibby et al. 2000). To identify the waterbirds, transects were simultaneously monitored by two observers using binoculars (10 × 50) and the Narosky and Yzurieta (2003) field guide was used. We identified the waterbird species seen or heard (species of the non-passerine order, which are totally or partially dependent on wetlands) and their abundance was recorded.

The species of waterbirds, registered in this study, were classified in guilds according to their foraging behaviour and diet (López de Casenave and Filipello 1995; Echevarria et al. 2014). The conservation status of the species was determined at a global level, using the International Union for Conservation of Nature's Red List of Threatened Species (IUCN 2020) and at a national level, the report of Ministerio de Ambiente y Desarrollo Sustentable y Aves Argentina (MAyDS and AA 2017). The scientific nomenclature of birds follows the classification of the American Ornithological Society (Remsen et al. 2020).

Statistical analysis

The abundance of each species was expressed as individuals per square kilometre (ind/km²). We used a non-metric multidimensional scaling analysis (NMDS) to

define the general differences in the structure and composition of bird assemblages between seasons. Ordinations were carried out using quantitative data (abundance, Bray-Curtis Index). We used an analysis of similarity (ANOSIM) to evaluate significant differences in the community structure between seasons. The statistical significance of the ANOSIM was assessed with a permutation test at a significance level of $\alpha = 0.05$. We used a similarity percentage analysis (SIMPER) to determine the contribution made by species and guilds to distinguish differences in the quantitative structure of the community between seasons (Clarke 1993). The Wilcoxon signed-rank test was conducted to determine whether or not there were significant differences in abundance of species and guilds.

To compare the taxonomic diversity between seasons, diversity profiles were obtained using the family of diversity indices proposed by Chao and Jost (2015). This method is based on the statistical estimation of the Hill numbers of any order $q \geq 0$. This estimator includes the three most used species diversity profiles (qD): species richness ($q = 0$), Shannon Diversity ($q = 1$, the exponential of Shannon Entropy) and Simpson Diversity ($q = 2$; the inverse of Simpson Dominance). These diversity estimators are designed to take into account the effect of undetected species in the samples, removing most of the biases associated with the empirical profiles. Estimated values of the order 0, 1 and 2 were analysed (Moreno et al. 2017). The analyses were performed using the *vegan*, *iNEXT* and *ggplot2* packages, for R version 4.0.1 (R Development Core Team 2020).

Results

During the sampling period, we detected a total of 38 species of birds from 11 families and seven orders (see Suppl. material: Table S1). The most representative families in relation to the number of species were Anatidae (37%), Ardeidae (16%), Rallidae (13%) and Podicipedidae (10%). We recorded a total of 1875 individuals of 33 species of waterbirds during the summer and 677 individuals of 29 species during the winter. The waterbird assemblages revealed grouping detections according to the season from axis 1 (Stress = 0.13; Fig. 1) and they were different in composition and abundance of species (ANOSIM, $R = 0.35$; $p < 0.01$). *Phoenicopterus chilensis* was the only species registered as nationally threatened (vulnerable; MADyDS and AA 2017) and globally threatened (near threatened; IUCN 2020).

The dissimilarity between both seasons was 73.84% and 13 species of birds contributed to 90% of the existing dissimilarity (Table 1). The bird species which contributed most to the assemblages in percentage were *Plegadis chihi* (20.8%), *Himantopus mexicanus* (20.7%), *Anas flavirostris* (12.5%) and *Fulica armillata* (10.8%). In total, five species of waterbirds increased their abundance significantly during the summer compared to the winter (Table 1): *Himantopus mexicanus* ($W = 2.20$; $p < 0.01$), *Netta peposaca* ($W = 1.58$; $p < 0.05$), *Fulica rufifrons* ($W = 1.99$; $p < 0.01$), *Egretta thula* ($W = 2.01$; $p < 0.05$) and *Butorides striatus* ($W = 1.58$; $p < 0.05$).

Table 1. Average abundance of waterbird species (ind/km²) and its contribution to the average dissimilarity between seasons (summer and winter) in Laguna del Viborón, Mendoza, Argentina. The species are arranged in descending order according to the percentage contribution of the SIMPER analysis. Diss. = Average dissimilarities; Cont. % = percentage contribution; % acum. = cumulative distribution. *indicates significant differences between seasons: ** = $p < 0.01$; * = $p < 0.05$ (Wilcoxon signed-rank test).

Species	Average abundance		Diss.	Cont. %	% acum.
	Summer	Winter			
<i>Plegadis chihi</i>	498.0	41.5	15.39	20.84	20.84
<i>Himantopus mexicanus</i> **	549.0	37.5	15.34	20.78	41.61
<i>Anas flavirostris</i>	347.0	356.0	9.25	12.53	54.14
<i>Fulica armillata</i> **	345.0	93.0	8.01	10.85	64.99
<i>Netta peposaca</i> *	236.0	2.8	4.75	6.43	71.42
<i>Anas georgica</i>	140.0	104.0	4.09	5.54	76.97
<i>Dendrocygna viduata</i>		106.0	2.22	3.00	79.97
<i>Fulica rufifrons</i>	86.8	2.8	2.20	2.99	82.95
<i>Egretta thula</i> *	51.3	15.2	1.58	2.14	85.09
<i>Cygnus melancoryphus</i>	30.5	1.3	1.49	2.02	87.11
<i>Calidris bairdii</i>	45.8		1.33	1.81	88.91
<i>Ardea alba</i>	21.0	15.2	0.77	1.04	89.95
<i>Podilymbus podiceps</i>	23.2	23.7	0.72	0.97	90.92
<i>Fulica leucoptera</i>	30.0	1.3	0.71	0.96	91.88
<i>Spatula cyanoptera</i>	6.3	25.0	0.64	0.87	92.75
<i>Butorides striata</i> *	18.7	1.3	0.61	0.83	93.58
<i>Rollandia rolland</i>	18.2	15.3	0.53	0.72	94.30
<i>Phalacrocorax brasiliensis</i>	10.3	8.3	0.49	0.66	94.96
<i>Anas bahamensis</i>	4.2	16.7	0.45	0.61	95.57
<i>Podiceps major</i>	11.8	13.8	0.41	0.56	96.13
<i>Pardirallus sanguinolentus</i>	16.7	13.7	0.41	0.55	96.68
<i>Vanellus chilensis</i>	24.3	14.0	0.40	0.55	97.23
<i>Coscoroba coscoroba</i>	11.8	5.5	0.39	0.53	97.76
<i>Heteronetta atricapilla</i>	11.2	2.8	0.24	0.33	98.09
<i>Tringa flavipes</i>	6.3		0.23	0.31	98.40
<i>Nycticorax nycticorax</i>	6.2		0.18	0.25	98.64
<i>Dendrocygna autumnalis</i>		8.3	0.18	0.24	98.88
<i>Spatula versicolor</i>	1.3	4.2	0.13	0.18	99.06
<i>Phoenicopterus chilensis</i>		4.2	0.13	0.17	99.23
<i>Ardea cocoi</i>	1.3	2.8	0.12	0.16	99.39
<i>Podiceps occipitalis</i>		2.8	0.10	0.14	99.53
<i>Tringa melanoleuca</i>	3.5		0.08	0.11	99.64
<i>Gallinula galeata</i>	2.7		0.08	0.11	99.75
<i>Ixobrychus involucris</i>	2.7		0.07	0.09	99.84
<i>Chroicocephalus serranus</i>		1.3	0.04	0.06	99.90
<i>Spatula platalea</i>	1.3		0.04	0.05	99.95
<i>Mareca sibilatrix</i>	0.7		0.02	0.02	99.98
<i>Oxyura vittata</i>	0.7		0.02	0.02	100.00

The registered species belonged to a total of eight trophic guilds (Table 2; see Suppl. material: Table S1) and were different in composition and abundance of species between the seasons (ANOSIM, $R = 0.48$; $p < 0.01$). The dissimilarity of the assemblage in relation to the guilds between both seasons was 63.67%. The guild of invertivorous waders ($W = 1.99$; $p < 0.05$) and omnivorous waders ($W = 1.99$; $p < 0.05$) had a greater abundance of individuals during the summer ($p < 0.05$) and contributed

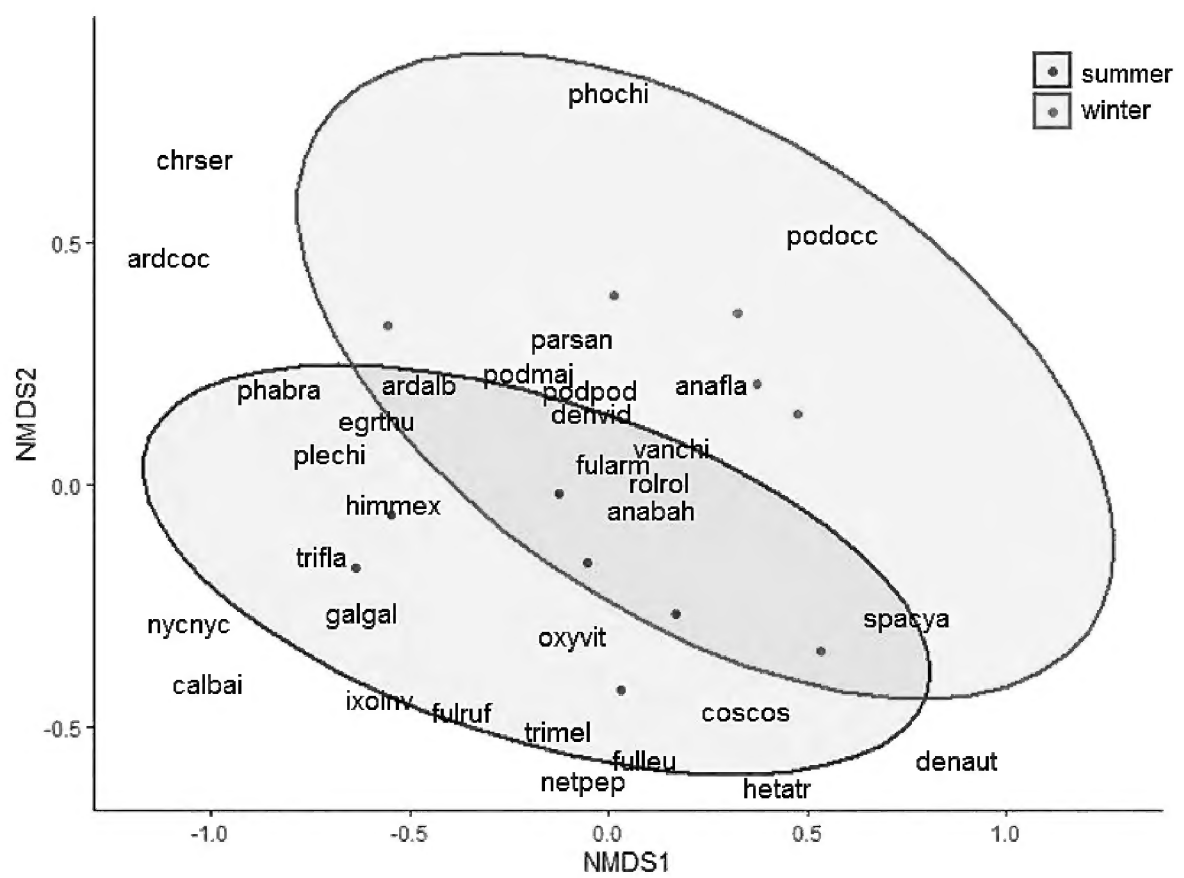


Figure 1. Ordination diagram using a non-metric multidimensional scaling (NMDS) of the waterbird assemblages in two seasons (summer and winter), in Laguna del Viborón, Mendoza, Argentina. Waterbirds: anabah = *Anas bahamensis*, anafila = *Anas flavirostris*, ardalb = *Ardea alba*, ardcoc = *Ardea cocoi*, calbai = *Calidris bairdii*, chrser = *Chroicocephalus serranus*, coscos = *Coscoroba coscoroba*, denaut = *Dendrocygna autumnalis*, denvid = *Dendrocygna viduata*, egrthu = *Egretta thula*, fularm = *Fulica armillata*, fulleu = *Fulica leucoptera*, fulruf = *Fulica rufifrons*, galgal = *Gallinula galeata*, hetatr = *Heteronetta atricapilla*, himmex = *Himantopus mexicanus*, ixoinv = *Ixobrychus involucris*, netpep = *Netta peposaca*, nycnyc = *Nycticorax nycticorax*, oxyvit = *Oxyura vittata*, parsan = *Pardirallus sanguinolentus*, phabra = *Phalacrocorax brasiliensis*, phochi = *Phoenicopterus chilensis*, plechi = *Plegadis chihi*, podmaj = *Podiceps major*, podocc = *Podiceps occipitalis*, podpod = *Podilymbus podiceps*, rolrol = *Rollandia rolland*, spacya = *Spatula cyanoptera*, trifla = *Tringa flavipes*, trimel = *Tringa melanoleuca*, vanchi = *Vanellus chilensis*.

Table 2. Abundance (mean ± standard error) and species richness (S) of waterbirds during two seasons (summer and winter), based on the guilds according to their foraging behaviour and diet, in Laguna del Viborón, Mendoza, Argentina. The guilds are arranged in descending order according to the percentage contribution of the SIMPER analysis. Diss. = Average dissimilarities; Cont. % = percentage contribution; % acum. = cumulative distribution. *indicates significant differences between seasons (Wilcoxon signed-rank test; *p* < 0.05).

Guilds	Summer				Winter				Diss.	Cont. %	% acum.
	ind/km ²		S (%)		ind/km ²		S (%)				
Invertivorous wader *	187.9	± 46.2	6	(18.2)	15.5	± 6.1	3	(10.3)	35.8	53.0	53.0
Surface-feeding herbivores	100.8	± 34.6	10	(30.3)	58.3	± 19.8	9	(31.0)	15.5	23.0	76.0
Surface-feeding omnivores	35.1	± 30.0	5	(15.2)	20.8	± 19.7	5	(17.2)	7.6	11.2	87.2
Omnivorous wader *	16.9	± 5.8	7	(21.2)	6.9	± 4.0	5	(17.2)	3.1	4.6	91.8
Piscivorous diver	11.3	± 2.8	3	(9.1)	12.2	± 5.3	4	(13.8)	2.4	3.6	95.3
Omnivorous diver	9.4	± 3.8	2	(6.1)	7.6	± 3.3	1	(3.4)	1.9	2.8	98.2
Filter-feeding omnivores					4.2	± 4.2	1	(3.4)	0.9	1.4	99.5
Omnivorous plunge-divers					1.4	± 1.4	1	(3.4)	0.3	0.5	100.0

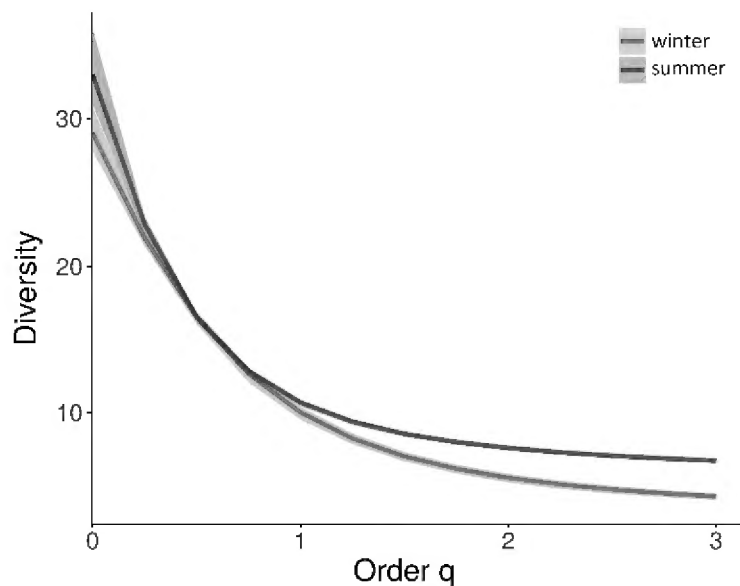


Figure 2. Estimated diversity profiles of the waterbird assemblages of Laguna del Viborón, Mendoza, Argentina, during two seasons (summer and winter).

together with the surface-feeding herbivores to the 87% dissimilarity of the total assemblage between seasons (Table 2). In both seasons, most of the species belonged to surface-feeding herbivores, omnivorous and invertivorous waders in summer and to omnivorous waders and surface-feeding omnivores in winter (Table 2).

The diversity profiles showed a significant difference between the seasons for all indices (qD) and the summer was the season with the greatest diversity (Fig. 2). For the summer, the diversity of zero order was $^0D = 33 \pm 1.42$ (30.98–36.15, with 95% Confidence Intervals), the diversity of first order $^1D = 10.57 \pm 0.11$ (10.32–10.76) and the diversity of second order $^2D = 7.46 \pm 0.09$ (7.31–7.64). For the winter, it was $^0D = 29 \pm 1.01$ (27.77–30.76), $^1D = 9.90 \pm 0.23$ (9.41–10.28) and $^2D = 5.42 \pm 0.16$ (5.11–5.70).

Discussion

Waterbirds assemblages in Laguna del Viborón showed seasonal variation in their structure. These changes are reflected both in the variation of individual abundance and in the species richness between summer and winter. These differences can be related mainly to seasonal variation in food resource availability, as well as seasonal movements patterns associated with migration and reproductive behaviour (Ronchi-Virgolini et al. 2013; Kopij and Paxton 2018). The seasonal variation in waterbirds' abundance has been documented in wetland located at high latitudes (Pedrana et al. 2018) and altitude (Caziani et al. 2001) in Argentina.

The increase in richness and abundance of certain guilds during the summer is related to the increase in food resources. Summer temperatures and the increased water flow favour the development of plant biomass in wetlands (Rivera et al. 2017). Thus, there is an increase in the supply and availability of food resources for birds, mainly of invertebrates, fish and plants (Minckley et al. 2013). Therefore, there are species which are favoured: herbivores, such as coots (*Fulica spp.*) and invertivorous and omnivorous waders, such as ibises and herons, respectively (Safran et al. 2000). Moreover, richness during the summer is also increased by the arrival of

invertivorous waders which migrate from the Nearctic (i.e. *Calidris bairdii* and *Tringa* spp.; Blanco et al. 2006). Moreover, the continental location of the wetland may explain the occurrence of certain species which make annual regional movements. This is due to the fact that, in the central Monte Desert, the latitudinal movements of southern (Patagonia) and northern bird populations converge in relation to seasonal changes (Chesser 1994; Jahn et al. 2020). In turn, this site is also a refuge for high Andean species, such as the Andean gull, which descend to lower altitude lands during unfavourable weather conditions in winter (e.g. snowfalls; Barcante et al. 2017).

In addition, the increase in the density of gregarious species is also due to the reproductive season. Inland wetlands in arid areas show a marked increase in the density of individuals during the reproductive season, in response to the hydrological cycle (Blendinger and Alvarez 2002; Alvarez et al. 2018). This is mainly due to the congregation of gregarious species in large flocks which usually concentrate in wetlands for breeding (Romano et al. 2005). Species such as *Netta peposaca* make seasonal migrations and increase significantly their abundance on the site during the summer (Antas 1994). The congregation of common species in large flocks (i.e. those with a large number of individuals compared to species of the same taxonomic group), such as ibises, coots and lapwings, may have a strong influence on the wetland dynamics. This is due to the fact that common bird species are key elements of the biomass, structure, functioning of the ecosystems and, therefore, the supply of ecosystem services (Sekercioglu 2006; Gaston et al. 2018). Although we did not carry out reproductive biology studies, we recorded nests and nestlings of *Fulica* spp., which were very abundant species during the summer.

The wetland may represent a resting and roosting site for individuals foraging in agricultural areas. Ibises, lapwings, herons and shorebirds usually move during the day in search of foraging sites (Lorenzón et al. 2017). Part of the population of these species may use the wetland as a resting and roosting place and move during the day to irrigated crops, using agro-ecosystems for feeding (Czech and Parsons 2002). There are certain anatids (*Anas flavirostris* and *A. georgica*) which also have a greater breadth of trophic niche and plasticity to exploit resources in agricultural areas, such as irrigation canals (Fox et al. 2017). At this site, such types of behaviour may be facilitated because the wetland is within an agricultural matrix which probably provides a sustained food supply (Contreras and Fernández 1980; Prieto and Rojas 2012). Waterbird use of irrigated agricultural areas as feeding sites has increased as natural wetlands continue to decline worldwide (Kingsford and Thomas 2004; Bellio et al. 2009).

In summer, the increase in water volume has an influence on a landscape scale, producing an increase in flooded areas. Flooded areas provide new habitats for waterbirds (Kushlan 1986). Water level fluctuations influence habitat physical structure (e.g. vegetation zonation), food availability and accessibility and presence of resting and breeding sites (Clausen 2000; Maleki et al. 2016). Long periods of flooding increase prey production (e.g. fish and aquatic macro-invertebrates; Ruetz et al. 2005). Nevertheless, in the last decade, water level shows a negative trend resulting in severe droughts (Masiokas et al. 2019), which may affect the dynamics of

the relict wetlands in the central Monte Desert (Benzaquen et al. 2017). The water deficit in wetlands of arid regions could cause negative cascading effects associated with widespread mortality of aquatic organisms (Ruetz et al. 2005), salinisation of soils (Rubio 2018) and eutrophication phenomena (Brinson and Malvárez 2002).

The decrease in the water level during winter favours certain species which forage in shallow waters (Dimalexis and Pyrovetsi 1997). Drought periods and water level fluctuations differentially affect bird species (Blendinger and Alvarez 2002; Vanausdall and Dinsmore 2019). Thus, during the winter, flamingos benefit from the decrease in the water level, since there are more foraging opportunities (Alvarez et al. 2018). Although this wetland is not the habitat preference of flamingos, it may constitute a resting place during their migratory movements (Caziani et al. 2007). Moreover, anatids of the genus *Dendrocygna* were recorded during the winter, species which occasionally visit the area (Martínez et al. 2009; Zanotti 2020). These birds usually forage in shallow wetlands and may arrive at the site due to migratory movements which deserve more consideration (Antas 1994).

The wetland is under the management by Cristobal Colón Fishing Club. It controls activities, such as illegal hunting and visitor access and they also divided the wetland into sectors, maintaining an area where people are not allowed to enter and this would be beneficial for birds (Cardoni et al. 2008). This could explain why this wetland has maintained its richness over time, since we register 10% more waterbird species than Contreras and Fernández (1980). Furthermore, the lack of identification of other species by records in the site (Contreras and Fernández 1980) can be limited to the sampling effort and also to the transformation of perilacustrine areas into cultivated lands (Rubio 2018). Human presence and habitat degradation affect the occupation of specialist species which have more habitat requirements, such as shorebirds (Blanco et al. 2006; Brandis et al. 2018). In addition, it has been found that the lack of management by the government regarding the conservation of wetlands led to the loss of waterbird diversity (Amano et al. 2018). Therefore, the current role of the social club may be positively significant in the conservation of waterbirds, given the lack of government policies for a sustainable management of the site.

Conclusions

The seasons showed different waterbird assemblages, with greater diversity during the summer. Guild structure was also different between seasons. Invertivorous waders and omnivorous waders showed greater abundance of individuals during the summer and, together with surface-feeding herbivores, they were the guilds which contributed most to the dissimilarity of assemblages. The structural characteristics and the environmental heterogeneity of the site are reflected in the presence of species with different habitat requirements. This can be observed in the presence of birds which make use of deep-water wetlands (*Cygnus melancoryphus*), shorelines (shorebirds), environments with dense vegetation (rails) and birds with specialist foraging behaviour (*Phoenicopterus chilensis*), amongst others.

The relict wetlands of the central Monte Desert, such as the Laguna del Viborón, still have a high diversity of species in relation to its surface area. In addition to functioning as wildlife refuges, they act as biological corridors to high-productivity wetlands which have large waterbird populations. Nevertheless, the wetlands which concentrate a large part of the populations of certain bird species (Alvarez et al. 2018) are sometimes affected by intense disturbances (e.g. pollution, drainage and drought; Abrahan 2019); therefore, the wetlands with smaller surface area, such as Laguna del Viborón, become relevant, functioning as a network of alternative emergency refuges. Due to the fact that the central Monte Desert in Argentina has lost almost the entire surface area of wetlands, relict wetlands should be a conservation priority for government agencies. To ensure the conservation of the waterbirds of Laguna del Viborón, integrated management plans should be implemented with an approach based on multiple spatial scales and temporal variability, allowing the protection of both the water resource and the different types of habitats for birds.

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Supplementary material 1

Table S1

Authors: Ever Tallei, Analía Benavidez, Alejandro Schaaf, Pablo Isola, Marcelo Zanotti

Data type: Species composition table (doc. file)

Explanation note: Composition of the (non-passerines) waterbird assemblage and their foraging behaviour and diet, recorded in Laguna del Viborón, Mendoza, Argentina, during two seasons (summer and winter).

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